The Potential Contribution of Waste Management to a Low Carbon Economy

Technical Appendices

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1.Introduction

These Appendices provide background information in support of the data in the main report. They consider the emissions from different waste prevention and management activities to climate change. The focus is on the impacts per tonne of the waste material being prevented / managed.

1.1 General Considerations for Waste Treatment Systems

When considering the greenhouse gas impacts of waste treatment systems – for either organic or residual waste - the following issues need to be considered:

- Direct emissions from the treatment process itself;
- Emissions associated with energy used within the treatment process; and
- The emissions which are avoided as a result of materials use (avoiding primary materials use), energy generation, and/or the benefits associated with the use of outputs, such as compost, that result from the treatment process.

The Appendices on source segregated organic waste and residual waste treatment systems therefore discuss the greenhouse gas emissions impacts on this basis.

2.Prevention, Re-use and Dry Recycling

In considering the climate change impacts of waste prevention initiatives, a distinction is made between:

• Activities that reduce the amount of material consumed without increasing the consumption of another type of material, such as light-weighting of single use packaging, or avoiding the wastage of food through judicious purchasing decisions. Benefits of these activities can be considered through data on the impacts of producing the materials that are the target of the activity.

• Initiatives where the reduction in the consumption of one type of material results in the increased consumption of another type of material. Here, emissions reductions may still be seen, but are often more difficult to quantify. Examples include swapping from single use plastic carrier bags to long life plastic bags, bags made from textiles, or single use paper bags.

The benefits of waste prevention relate in part to the type(s) of material(s) whose consumption is being avoided. Also, for reasons explained below, if the material whose consumption is avoided is derived mainly from recycled sources, the benefits of avoiding consumption might be lower than in the case where the material is derived mainly from primary sources.
Key factors determining the impacts of these measures are considered below:

1) Prevention
Key factors are:

a. The materials inputs used in producing the goods, or packaging, which is being prevented;

b. The amount of energy used in the process used to produce the related materials, and the type of energy used in the process. Production is more energy-intensive for some materials than others, and so the impacts of production vary by material. For reasons explained in the context of recycling below, the mix of primary and secondary materials used in the production process will influence the amount of energy that is required in production. Typically the use of recycled content results in a reduction in production emissions, so the proportion of recycled content used in the production process is also important. This varies across different countries, with Europe being more advanced than elsewhere given the recycling targets contained within the Directives. As the proportion of recycled content increases, so the benefits of the source reduction initiatives may be expected to be reduced.

c. Electricity consumption is more carbon-intensive than heat production per kWh of energy, so the type of energy consumed in the production process is also important, as is the carbon intensity of the source of heat or electricity.

d. Since different countries use different sources of fuel for energy generation, the country where manufacture takes place may also be important, especially where, for example, electricity is concerned. In many countries, policies aimed at decarbonising energy supplies will reduce the impacts from many production processes over time. Decarbonisation plans are relatively more advanced for electricity production, and so in the short to medium term, the impact is anticipated to be greatest on those production processes that are more reliant upon electricity consumption.

2) Reuse
For reuse, similar factors to those considered above for waste prevention are relevant. However, there are additional factors which are of relevance, with the key issue being how the emissions associated with the cycle(s) of reuse compare with this situation which would have prevailed without the reuse activity.

a. Where the nature of the reused product is different to that of the single trip product it replaces (for example, glass bottles designed for several reuse cycles may be heavier than single trip bottles as they are designed to be handled and reused many times), then the relative energy intensity of the production of the reused product and the displaced single trip product are important. It also becomes important to know how many times the reusable product can be reused before it either breaks, or loses its functionality for another reason (other things being equal, the more times the reused product can be utilised, in general, the better);

b.For some reused goods, the good consumes energy as it is used. It then becomes important to understand the relative energy use of the reused product relative to what would otherwise have happened in the absence of reuse. For example, in the absence of reuse options, would consumers purchase new, more efficient products, or would they not have purchased them at all?

c. There may also be energy used in the process by which goods or packaging are prepared for reuse (for example, in washing of reusable nappies).

3) Recycling

Where recycling impacts are concerned, the main impacts relate to the greenhouse gas impacts of:

a. The change in emissions associated with the changes in collection and sorting systems (including bulking and haulage), though these tend to be relatively small;

b. The rate at which the materials collected for recycling substitute for primary materials (so, for example, the benefit will be greater the closer the rate of substitution is to 100%);

c. The change in the amount, and source, of energy used when materials are produced using secondary materials instead of primary ones. There are large reductions in GHG emissions in the case of the recycling of metals, a significant reduction in the case of recycling of plastics, and a smaller reduction in the case of recycling of glass, or wood;

d. The location for reprocessing of the secondary material and the location of production of the primary material for which the secondary material substitutes. As markets for primary and secondary materials are global ones, obtaining representative emissions factors for a specific country is a challenging task.

2.1 Waste Prevention

Waste prevention impacts for the situation where no material substitution occurs can be considered through the avoided manufacturing impacts. Sources of information in this respect include the Scottish Carbon Metric, which reviewed the data on production in Europe and China, as well as the life cycle databases such as Ecoinvent. Selected data is presented in Table 2 -1. We used the data from the Scottish Carbon Metric (SCM) in our analysis as this is both more recent and more consistent with the recycling data presented in Section 2.2 – noting that in some cases (such as for steel) the source of the information in the SCM is, in fact, the Ecoinvent database.

The data in the above table on production emissions are applicable to prevention initiatives, such as a reduction in the amount of packaging material used, or initiatives aimed at tackling food waste. Waste prevention activities are clearly much wider in scope than this. However, as was indicated above, emissions savings resulting from some of these other initiatives are rather more difficult to quantify where one activity is being replaced by another. Table 2 -2 provides data on this type of action, which covers such initiatives as the use of real nappies (displacing the use of disposables).

Table 2-1: Data on Waste Prevention Impacts

	Avoided emissions, kg CO2 eq. per tonne	
Material	**Scottish Carbon Metric**	**Ecoinvent**
Paper / card	-893	-1,693
Plastic	-3,41	-1,948
Glass	-895	-874
Textiles	-21,148	-24,3
Steel	-2,937	-2,092
Aluminium	-12,96	-12,043
Food waste	-3,8	N/A
Discarded machines and equipment (includes WEEE)	-1,754	N/A
Others	-1,91	N/A
Mineral waste from construction and demolition	-12	-2

Notes:
1) These figures are indicative of the benefits of waste prevention initiatives where there is no displacement of one material by another as a result of the initiative
4) Data on food waste production emissions is derived from WRAP, whilst the other data is sourced from Zero Waste Scotland. Much of the data is taken from international datasets and is therefore felt to be applicable to European countries in general.
5) The data on textiles assumes some re-use as well as recycling.

Table 2-2: Impacts of Some Other Waste Prevention Initiatives

Initiative description	Indicative emissions impact	Source
Use of "real" nappies displacing disposable nappies	Suggested up to 40% emissions reduction over the period a child uses nappies (results are dependent on laundering assumptions, and do not assume the resale of the cloth nappies)	UK Environment Agency
Single-use HDPE carrier bags replaced with long-life plastic bags	-6 kg CO2 eq. per household	Sustainability Victoria
Replacing Styrofoam cups with refillable cups	-58 kg CO2 eq. over lifetime of refillable cup	Refiller
Textiles	-21,148	-24,3

Sources: Environment Agency (2008) An Updated Lifecycle Assessment Study for Disposable and Reusable Nappies; Sustainability Victoria (2007) Comparison of Existing Life cycle Analysis of Shopping Bag Alternatives; Refiller (2013) Lifecycle Assessment: Reusable Mugs vs. Disposable Cups WRAP (2011) Benefits of Reuse Case Study: Electrical Items; WRAP(2011) Benefits of Reuse Case Study: Domestic Furniture

2.2 Re-use

Table 2 -3 presents some data on the benefits of re-using items such as furniture and WEEE. Here, the climate change benefits are more difficult to evaluate on the basis of the impact per tonne, as this is often not the most appropriate way to consider the impacts. Per-tonne impacts are shown for the re-use of certain furniture items, however, and this shows that the impacts are more modest than might expected given the impacts associated with recycling some of the key components. This is partly because, in some cases, the subsequent owner of the re-used product would not otherwise have bought a new product. Such activities result in wider benefits to society – for example, low income households obtain furniture they would otherwise not be able to purchase – but they do not necessarily translate into substantial emissions savings. For this reason, the benefits associated with re-using office furniture are higher than those associated with similar domestic products, as in the case of the former, the purchase of a new product is much more likely to occur in the instance where no re-used items were available.

Table 2-3: Impacts of Some Re-use Activities

Initiative description	Indicative emissions impact	Source
Re-use of washing machine	-500 kg CO_2 eq. per tonne via a charity shop; 200 kg CO_2 eq. per tonne via a re-use network	WRAP
Re-use of televisions	-8,000 kg CO_2 eq. per tonne via a charity shop; -5,000 kg CO_2 eq. per tonne via a re-use network	WRAP
Re-use of sofa	-1,450 kg CO_2 eq. per tonne via a charity shop; -1,005 kg CO_2 eq. per tonne via a re-use network	WRAP
Re-use of dining table	380 kg CO_2 eq. per tonne via a charity shop; 760 kg CO_2 eq. per tonne via a re-use network1	WRAP
Re-use of office desk	-400 kg CO_2 eq. per tonne via a charity shop; -200 kg CO_2 eq. per tonne via a re-use network	WRAP
Re-use of office chairs	-3,000 kg CO_2 eq. per tonne via a charity shop; -2,600 kg CO_2 eq. per tonne via a re-use network	WRAP
Notes: The analysis undertaken here does not account for the biogenic carbon sequestered in the tables (which are made of wood) as a result of re-use, which would be expected to further increase the benefits from re-use.		

Sources: WRAP (2011) Benefits of Reuse Case Study: Electrical Items; WRAP (2011) Benefits of Reuse Case Study: Domestic Furniture

2.3 Data on Dry Recycling

2.3.1 Paper

Paper is typically the material recycled in the largest quantities, and as such the value attributed to paper recycling is of particular significance in this type of analysis. It will be seen, however, that there is considerable variation in the literature with regard to the benefit attributed to recycling the different types of paper products.
Although many European paper re-processors such as UPM have collated data on the energy use associated with their paper manufacturing processes, there remains a relative lack of recent datasets that can be used to calculate the benefits of paper recycling. UPM uses recycled fibre in most of its products, and thus the company has only limited data relating to the counterfactual (i.e. the manufacture of paper from virgin raw materials).
Paper and board collected for recycling are reprocessed into newsprint or packaging products. These two products are associated with two very different manufacturing processes; this is discussed below. We also provide a brief review of the literature surrounding the benefits of paper recycling and consider the impact of paper recycling upon carbon stocks in trees in this section.

2.3.1.1 Paper and board manufacturing processes

Emissions savings are dependent on the type of virgin fibre manufacturing process that is substituted by the inclusion of the recycled fibre.
There are two broad groups of virgin paper manufacture processes:

- Chemical pulping processes: This involves the removal of lignin from the wood. The process preserves fibre length which results in the manufacture of a stronger product. The removal of the lignin results in a lower yield of paper such that only 40-50% of the original wood is subsequently converted into usable fibre; as such the process is a relatively expensive one. However, typically most of the significant quantities of heat and electrical energy needed for the virgin manufacturing process can be supplied through the use of steam produced during the combustion of the lignin removed during the pulping. This type of process is typically used to manufacture cardboard (usually made up of three layers of very strong brown ‘kraft’ paper) and other packaging materials, and in the manufacture of other high quality paper products; and
- Mechanical pulping processes: In this case the lignin is not removed, so the fibre yield is very high. As such the manufacturing process is relatively cheap, despite the requirement for significant quantities of electrical energy – energy which is more likely to be supplied by an external, fossil fuel-based source than is the case in the chemical pulping process. However, the retention of the lignin results in a weaker product with less tensile strength that has a tendency to become yellowed and brittle over time. Newspaper is typically manufactured using mechanical pulping processes, as is the paper used in mass-market book manufacture.

Whilst relatively little fossil electricity is typically used for the manufacture of virgin cardboard, requirements are typically greater where board is reprocessed from collected recyclate. As such, the benefits associated with the recycling of fibre into packaging products are typically reduced in comparison to those of reprocessing fibre into newsprint. Recycling board results in a significant benefit in terms of biogenic CO_2 emissions, however, due to a reduction in bio-energy requirements associated with the recycling process.

2.3.1.2 Values from the literature

Table 2 -4 confirms impacts associated with recycling newsprint from a number of literature sources. The data presented in the table excludes the biogenic CO_2 emissions. The majority of the literature sources that consider the benefits associated with recycling paper do not separately identify changes in emissions of biogenic CO_2 that occur as a result of paper and cardboard recycling. However, ecoinvent includes these emissions as an information item in the full emissions inventory although the amounts are not included in the calculation of the GWP impact of recycling paper products.

The ecoinvent dataset is derived using the following assumptions, which compare two processes:

- Newsprint production with 0% DIP (deinked pulp from recycled paper). Under this process 2.5 tonnes of virgin wood product is used to create 1 tonne of newsprint and no pulp is used;
- Newsprint using DIP. Under this process 0.756 tonne of pulp from recycled paper offsets 1.4 tonne of virgin product to produce 1 tonne of newsprint; [1]
- This substitution results in a saving of 0.22 kg CO_2 equivalent per tonne of newsprint produced;
- Pulp production efficiency from newsprint is assumed to be 80%;
- When the materials substitution ratio is combined with the pulp efficiency, this results in a net climate change impact of -0.23 tonnes CO_2 equivalent per tonne of newsprint to the process.

The ecoinvent data uses information from 2000. Similar values have been generated in two more recent datasets – one of which (US EPA) is likely to be representative of a relatively high carbon energy mix, whilst the other (Raandal) is likely to better reflect a lower carbon energy mix.[2] More recent data cited in the table from the version of the SCM published in 2013 also apparently uses the ecoinvent database but no details were provided on the calculations.

In contrast, the newer version of the SCM published uses data from a number of sources looking at the recycling of paper in China; the methodology used here is not clearly stated. Values are similar to the earlier Prognos study, which provides much higher values but also gives very little information on the origins of the data used. However this SCM data is perhaps less appropriate for a European value as it appears to assume all the material is exported. Whilst this may be true for Scotland, data provided from European databases such as the Market Access Database suggests less export from the EU as a whole.

Table 2-4: Selected Values – Impacts of Recycling Newsprint

Cited source	Data source origin	Climate change impacts t CO_2eq. / t	Substitution of recycled vs virgin	Electricity source(s)
Ecoinvent (2000)	European average	- 0.23	0.8:1	Euro 2000
SCM (2013)1,2	ecoinvent	-0.34	Unknown	Unknown
SCM (2014)2	various	-0.88	Unknown	Unknown
Raandal (2009)	Norway	-0.2	0.85:1	Norway 2008
AEA 2001	Swedish study	-0.63	Unknown	EU mix 1996
US EPA 2006	US EPA	-0.21	0.94:1	US mix
Prognos / IFEU 2008	Not stated	-0.8	1% process losses (recycled fibre)	Not stated
Notes: 1 Source is also apparently ecoinvent (not clear which processes were used). 2 Generic data for paper / board rather than newspaper.				

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; USEPA (2006) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008; ecoinvent database; Raandal (2009) Klimaregnskap for avfallshåndtering; Zero Waste Scotland (2014) 2012 Updates to the Carbon Metric Technical Report, August 2014

1 The process also uses a further 1.1 tonnes of wood product

2 The substitution ratio for US EPA study was 94%, whilst that of the Norwegian Randaal study was 85%. See: USEPA (2006) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks; Raandal (2009) Klimaregnskap for avfallshåndtering

Table 2 -5 presents selected data from the literature on the climate change impacts associated with the recycling of board and corrugated card, all of which present the impact excluding the biogenic CO_2 emissions.

The ecoinvent figure uses a pan-European dataset provided by FEFCO in 2005, and is developed by comparison with two European corrugated card processes producing single wall card – one using fresh fibre and the other recycling fibre. A 1:1 substitution ratio is assumed in this case, as the background data by FEFCO suggests the total quantity of inputs is the same to both processes. This assumption is in line with that used in a later study by Raandal which gives higher benefits even with an energy mix that has a lower carbon intensity (which might be expected to reduce benefits). The 2006 US EPA study used a slightly lower substitution ratio but arrives at lower benefits despite a relatively high carbon energy mix. It can be seen that where the fibre is reprocessed into cardboard packaging, this has a much lower benefit than where biogenic CO_2 emissions are excluded. Benefits are, however, far greater where the biogenic CO_2 emissions are included, for the reasons outlined in Section 2.3.1.1.[3] Thus applying the same methodology the ecoinvent data indicates that biogenic CO_2 benefits of 1.4 tonnes CO_2 equivalent per tonne of corrugated card recycled.

Similar to the situation with newsprint, the ecoinvent data is the only one of the three to provide data on the other air pollutants (including the biogenic CO2 emissions) and the data appears to be roughly in line with those seen in the other relatively recent studies, as such, the values derived from this data source have been used in the model.

Table 2-5: Selected Values – Impacts of Recycling Board and Corrugated Card

Cited source	Data source origin	Climate change impacts t CO2eq. / t	Substitution of recycled vs virgin	Electricity source(s)
Ecoinvent	Europe	-0.01	01:01	EU mix 2005
AEA 2001 (EU)	Swedish study	-0.12	Unknown	EU mix 1996
US EPA 2006	US EPA	-0.01	0.93:1	US mix 2006
Raandal (2009)	Norway	-0.15	01:01	Norway 2008

Notes:
1 Source is also apparently ecoinvent (not clear which processes were used).
2 Generic data for paper / board rather than newspaper.

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, May 2002; WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006; Grant et al (2001) LCA of Paper and Packaging Waste Management Scenarios in Victoria, Report for EcoCycle Victoria; Paper Task Force (2002) Life cycle Environmental Comparison: Virgin Paper and Recycling Paper Based Systems, White Paper No. 3; ecoinvent database

2.3.1.3 Value used for Paper / Card Recycling

The value from the 2013 SCM has been used in the analysis. This is taken to be representative of a value combining the paper and card impacts, and where a proportion of the material is exported to China for reprocessing.

2.3.1.4 Carbon Stocks in Trees

Modelling originally carried out in 2002 by the US EPA (and retained in the 2006 version of the same study) included quite a sophisticated consideration of the US forest sector, and the implications of not harvesting forests as a result of paper recycling: [4]

3 Electricity from fossil sources is often required for the manufacture of board from recycled material, although the requirements of the virgin manufacture process are usually met through the combustion of lignin.
4 USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002

'When paper and wood products are recycled or source reduced, trees that would otherwise be harvested are left standing. In the short term, this reduction in harvesting results in a larger quantity of carbon remaining sequestered, because the standing trees continue to store carbon, whereas paper and wood product manufacture and use tends to release carbon. In the long term, some of the short-term benefits disappear as market forces result in less planting of new managed forests than would otherwise occur, so that there is comparatively less forest acreage in trees that are growing rapidly (and thus sequestering carbon rapidly).
Considering the effect of forest carbon sequestration on U.S. net GHG emissions, it was clear that a thorough examination was warranted for this study. The complexity and long time frame of carbon sequestration in forests, coupled with the importance of market dynamics that determine land use, dictated the use of best available models.'

The US EPA suggested that additional biogenic CO_2 emissions of 2.677 tonnes CO_2 equivalent per tonne of paper could be saved through this aspect of paper recycling. However, data presented elsewhere by the European paper and packing industries suggests that in Europe, relatively few trees are felled to make paper, and further indicates that only around 11% of the timber felled throughout the world is used to make this type of product: [5]

From a tree, big logs are used for timber. The branches cut to maintain trees healthy are used for paper making as well as residues from saw mills such as wood chips, are also raw material for paper…… Over the years, thinning operations weed out the weaker trees, but there is still a net gain in forest stocks - the Food and Agriculture Organization (FAO) indicates that there is an annual forest growth of 5 per cent in the northern hemisphere.

In addition, the European paper industry indicates that it supports certification as a way of documenting sustainable forest management, with certificates based on defined criteria issued by independent auditors making this verifiable by customers and consumers. Further European paper and packaging industries statistics suggest that 82% of forests owned by paper companies are certified in this way.
Of the datasets examined above, only the US EPA considered the potential impact in forestry-based carbon stocks, and their analysis has only considered the situation for the US. We have not included this impact in the current analysis, as a result of the uncertainties associated with this type of modelling and its application to current European forestry and paper manufacture practices.

2.3.2 Glass

The assumptions associated with both the closed-loop (glass recycled back into glass) and open-loop (glass used to replace other materials) glass recycling processes are described in this section.

2.3.2.1 Closed loop recycling processes

Table 2 -6 presents data from selected literature sources with regard to the climate change impacts of recycling glass. The WRAP study attributes relatively high benefits to recycling glass. WRAP's dataset includes some European studies from the 1990s (e.g. one study uses UK electricity fuel mix data from 1990, when around two thirds of the mix was from coal generation) and also includes one case study looking at reuse for which the benefits are much higher. However, the data from European Container Glass Federation (FEVE) – the most recent dataset – attributes a higher benefit from recycling than the WRAP dataset. [6]
For closed loop or 'remelt' glass recycling processes, the British Glass dataset included within WRATE indicates impacts to be -0.117 tonne CO_2 equivalent per tonne of material reprocessed for green glass, and -0.227 kg CO_2 equivalent for brown. Documentation

5 See http://www.paperonline.org/
6 The project team requested access to the full dataset from FEVE but no response was received to the request

provided with the software indicates the source of the data to be Berryman (a major UK glass re-processor) for the impacts associated with the reprocessing part of the system, whilst data from British Glass is used to assess impacts associated with the primary manufacture of the product. The analysis of the two together was provided by Enviros in 2004. There is a minor variation in the benefit associated with the processing of the different colours of glass. The average of the two British glass values is very similar to the more recent study published by Prognos which states their value was calculated assuming a feedstock that contained 75% cullet (no details of the source of their data were however provided). A similar value is also given in the more recent dataset from the SCM.

The study undertaken by Enviros in 2003 also attributes a greater benefit to closed loop glass reprocessing, and in this case the data source is also taken to be British Glass albeit that this study was undertaken the year before that which was used to determine the benefits in WRATE. It is not clear whether an updated dataset was used to calculate the benefits assumed by WRATE.

Documentation provided by WRATE suggests that the dataset was provided assuming a 1:1 displacement of virgin cullet by recycled glass. The rationale for this was that there is little in the way of losses through the recycling process. The EC report on the Best Available Techniques for glass reprocessing confirms that some virgin cullet is always required in the glass manufacturing process, suggesting the 1:1 displacement may be optimistic, although the report also indicated that substitution rates of over 90% were possible.[7] The data from the European Container Glass Federation (FEVE), on the other hand, was calculated assuming 1 kg of cullet displaces 1.2 kg raw materials resulting in a much higher benefit being seen for recycling glass than all of the other datasets.[8]

Although purporting to reflect the situation in Europe, the FEVE dataset may overstate the benefits associated with recycling given their very favourable substitution ratio. As such, we have used the value for brown glass from British Glass in our model. As is the case with the ecoinvent data, the use of the WRATE data point allows for consideration of the air quality impacts associated with recycling. Given that the value from this study is similar to the later va-

Table 2-6: Selected Values – Benefits of Recycling Glass (Closed Loop Processes)

Data source	Impacts (tonnes CO2 equivalent per tonne of glass recycled)
British Glass (from WRATE) (2004)	Green -0.117 / Brown -0.227
FEVE	-0.67
US EPA (2002/6)	-0.28
AEA (2001)	-0.29
WRAP (2006)	-0.44 average
Enviros (2003)	-0.31 UK / -0.29 overseas1
Prognos / IFEU (2008)	-0.18
SCM (2014)	-0.2
Notes: 1.Enviros study also used data from British Glass (as with WRATE)	

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; ERM (2006 a) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; ERM (2006 b) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste

7 IPPC (2013) Best Available Techniques (BAT) Reference Document for the Manufacture of Glass, JRC Reference Report
8 FEVE indicates that the dataset was developed using Italian data, although no detailed information on the methodology is publicly available
Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008; WRATE database; FEVE data available from http://www.feve.org/index.php?option=com_content&view=article&id=40%3Alca-1&catid=1&Itemid=32

lues from Prognos (and that the latter was intended to representative of benefits of recycling occurring in Europe), the use of the WRATE data is felt to be reasonable.

2.3.2.2 Open loop recycling processes

For open loop glass recycling processes, WRATE considers the benefits associated with the production of glasphalt, where crushed glass is mixed with aggregate and bitumen in a conventional asphalt plant. This is considered to result in climate change impacts of -0.021 tonnes of CO_2 equivalent per tonne of material reprocessed. No detailed information on the nature of the assumptions used within the study was provided.

The 2006 WRAP review looked at data from five studies. From these studies, the best performance associated with the open loop reprocessing of glass was -0.1 tonne CO_2 equivalent whilst the average was -0.01 tonnes. Although these values included the avoided disposal impacts, since these were associated with landfilling of inert material, this is not expected to have had a significant impact on the results.[9] The WRATE data therefore appears to be in line with that of the WRAP review.

2.3.3 Steel

Table 2 -7 presents data on the climate change benefits of recycling steel. In the case of steel recycling, WRAP's 2006 study found that the assumptions which had the highest influence on the results were those related to the interdependency of the steel waste handling system with the energy system of the surrounding technosphere – particularly with regard to the type of energy used within the primary and recycled scrap manufacturing systems. In this case, elements of both the primary manufacture and reprocessing system may occur outside Europe, and so the carbon intensity of electricity generation needs to be considered in a global context.

ecoinvent attributes an impact of -1.6 tonnes CO_2 equivalent per tonne of steel recycled. This is very similar to assumptions used in the recent SCM which was calculated from World Steel data (the latter based on analysis undertaken in 2009), although no detail is available in respect of the assumptions used in the calculation of their figure.[10]

Interestingly, ERM's 2006 report also uses ecoinvent data and but suggest much lower values. No details of the assumptions regarding the location of mining and manufacturing operations were given in the more recent ERM studies, and so it is not clear

Table 2 7: Impacts of Recycling Steel from Various Literature Sources

Data source	Impacts (tonnes CO2 equivalent per tonne of glass recycled)
SCM / World Steel (2013)	-1.8
ecoinvent (2003/6)	-1.6
US EPA (2002/6)	-1.79
AEA (2001)	-1.52
WRAP (2006)	Average -1.34 (of landfill scenarios)
ERM (2006 a)	-0.43
ERM (2006 b)	-0.58 – -0.83
Prognos / IFEU (2008)	-1.0

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; ERM (2006 a) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; ERM (2006 b) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008; ecoinvent database

9 WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

10 The project team requested access to the World Steel LCA dataset as part of this project but received no response to our request

whether the former were also considered to occur outside Europe. The figure from the database is, however, more in line with those of the older US EPA and AEA studies. The more recent Prognos study provides only an estimated value for the benefits of steel recycling, with their estimated figure being higher than that of ERM, but lower than the earlier AEA and US EPA values.

The use of the ecoinvent values is felt to be a representative estimate of the benefits associated with steel recycling.

2.3.4 Aluminium

Table 2 -8 presents other values from alternative literature sources. The data in the table confirms that

Table 2-8: Impacts of Recycling Aluminium from Various Literature Sources

Data source	Impacts (tonnes CO2 equivalent per tonne of aluminium recycled)
ecoinvent	-10.7
US EPA (2002/6)	-15.07
AEA (2001)	-9.20
European Aluminium Association (2008)	-9.17
SCM (2014) / EEA	-9.99
ERM (2006 a)	-11.60
ERM (2006 b)	-12.30 – -13.10
Prognos / IFEU (2008)	-11.1

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; ERM (2006 a) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; ERM (2006 b) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; European Aluminium Association (2008) Environmental Profile Report for the European Aluminium Industry: Life Cycle Inventory Data for Aluminium Production and Transformation Processes in Europe, April 2008; USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008; ecoinvent database; Zero Waste Scotland (2013) The Scottish Carbon Metric: Technical Report, October 2013

ecoinvent attributes an impact of -10.7 tonnes CO_2 equivalent per tonne of aluminium recycled. This is lower than the benefit calculated in several of the other sources reviewed, such as the Prognos, US EPA and the ERM analyses (as was the case with steel, ERM also used data from ecoinvent) – but is slightly higher than the EAA estimate. The most recent data from the SCM uses information from the EEA and updates the production data from the earlier dataset in 2008. We have chosen to use the SCM data in the model as the study is indicates this is based on average European data, and the data is the most recent of those reviewed.

2.3.5 Plastics

Table 2 -9 provides values from selected literature sources relating to the impacts associated with the recycling of dense plastic. The Table includes data on the relative benefits of recycling the different plastic polymers where these were identified in the literature.

Data provided by the Association of Plastics Manufacturers in Europe (APME) taken from WRATE attributed an impact of -1.04 tonnes CO_2 equivalent per tonne of mixed plastic recycled, whilst the recycling of plastic bottles was considered to result in climate change impacts of -1.15 tonnes CO_2 equivalent per tonne of plastic bottle recycled.

In the case of plastics recycling, the 2006 WRAP review identified a number of sensitive assumptions from the range of studies it had considered. Particularly important were the following:

- whether washing or cleaning of the material was required – where this was the case, benefits were decreased (as a result of the use of hot water); and
- whether the recycled material was assumed to substitute (on a tonne for tonne basis) virgin material of the same kind. In cases where the quality / grade of the recovered plastic implied a less favourable substitution ratio (worse than 1:1), the scenarios dealing with this issue demonstrated that a ratio of 1:0.5 was about the break-point at which recycling and incineration with energy recovery were environmentally equal.

Table 2-9: Selected Values – Impacts of Recycling Dense Plastic

Data source	Impacts (tonnes CO2 equivalent per tonne of aluminium recycled)
Association of Plastics Manufacturers in Europe (in WRATE)	Mixed plastic -1.04
	Bottle plastics -1.15
US EPA (2002/6)	HDPE -1.40
	LDPE -1.71
	PET -1.55
AEA (2001)	HDPE -0.53
	PET -1.80
APME (2005)	HDPE -1.90
WRAP (2006)	Average -1.08 (of landfill scenarios)1
ERM (2006 a)	-2.32
ERM (2006 b)	1.82 (lumber) / -0.85 closed loop
Prognos / IFEU (2008)	-0.16 – -1.72
SCM (2013)	-0.578
Franklin Associates (2010)	PET -1.98 HDPE -1.2
Notes: 1. Unlike the other studies referenced above, WRAP's values included the benefits associated with avoided residual treatment; these are, however, likely to be minimal for landfilled plastic. 2. Depending on production process and polymer mix	

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; ERM (2006 a) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; ERM (2006 b) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; APME data cited here from http://www.plasticseurope.org ; USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008; WRATE database; Zero Waste Scotland (2013) The Scottish Carbon Metric: Technical Report, October 2013; Franklin Associates (2010) life cycle inventory of 100% postconsumer HDPE and pet recycled resin from postconsumer containers and packaging, Report for The plastics division of the American chemistry council, inc., July 2010

WRAP's analysis found, however, that the results were relatively less sensitive to the type of polymer being recycled. The more recent data by Prognos indicated, however, that there was some variation across the different polymers, although since some variation in manufacturing process was also considered across the datasets, the variability that might be attributable to the polymer is difficult to determine. The latter study suggested a relatively large range in the potential benefits from plastic recycling than is seen in the other studies presented in the Table.

The data in the Table suggests that the dataset from the APME taken from WRATE attributes a lower benefit to plastic recycling than many of the literature sources contained within the Table. The values for benefits of bottle recycling are calculated assuming a composition of bottles containing different plastic polymers. A closer inspection of the value attributed to the recycling HDPE bottles within WRATE confirms that impacts of -1.182 tonne of CO_2 equivalent per tonne of recycled material are anticipated – a much lower benefit than that attributed by APME in the data published on the Plastics Europe website despite the data apparently originating from the same source at around the same time. The reason for this substantial differential is not clear.

More recent data on primary production of the various polymers is available from the successor to the APME, Plastics Europe but this data does not include information on the recycling process.[11] However, recent data is available for the USA for PET and HDPE, which again suggests the values vary by polymer.[12] Where this data is compared with the primary production data from Plastics Europe, it can be seen that impacts of primary PET production in Europe are much lower than those of the US, although the figure for HDPE is comparable. This suggests that the benefits of PET recycling in the European context may be lower than that indicated for US. No data is available on the impacts of recycling the polymers PP and PS, commonly found in the mixed plastics waste fraction, although the Plastics Europe data indicates that the impacts of producing the primary polymer are less than those of PET.

These values are based on closed loop recycling processes. Other sources have suggested that significant quantities of plastic is being recycled using open loop processes, which result in lower environmental benefits; as such, the recently revised data provided in the SCM – which took such an approach – indicates much lower benefits of 0.587 tonnes CO_2 equivalent per tonne of material recycled. [13]The proportion of material recycled via such processes is likely to vary between countries and over time.

The Franklin Associates study is likely to overstate the benefits of recycling PET, and may also overstate benefits of recycling plastics in general, given that the values assume all materials are recycled using a closed loop process. Franklin Associates also do not give values for the mixed plastics. The Plastics Europe data suggests that the impacts of manufacturing primary PP and PS are considerably lower than that of PET, suggesting that the benefits of recycling the material are likely to be more in line with those of HDPE. Given this, the APME data contained within WRATE is therefore again taken as being reasonably representative, given the considerable uncertainty that surrounds the calculation of the benefits associated with recycling this type of material. The use of this dataset similarly allows for a consideration of air quality impacts alongside the climate change benefits.

2.3.6 Textiles

Table 2 -10 presents data from a number of literature sources that have examined the potential climate change impacts associated with recycling textiles. Textiles were not considered in the 2006 WRAP review – reflecting, in part, the lack of detailed analysis that had been undertaken in this field at the time of the publication of their study.
It is clear that the benefits associated with recycling textiles vary enormously, with some of this variation

11 See: http://www.plasticseurope.org/plasticssustainability/eco-profiles.aspx
12 Values were calculated based on US energy mix, and assuming a 20% loss rate in the recycling process relative to the virgin manufacturing process.
13 Some supporting information on the validity of such an assumption in the UK was published by ERM in 2010, which suggested that 30% of bottles were recycled via open loop processes. See: ERM (2010) Life cycle assessment of example packaging systems for milk, Report for WRAP, January 2010;

being dependent upon the type of fibres and the end use of the recovered material. The lower end of the range of the ERM (2006a) data, for example, is said to relate to the recycling of poor quality material into rags or fillers. The upper end, however, indicates benefits of 7 tonnes CO_2 equivalent per tonne of material recycled.

Table 2-10: Impacts of Recycling Textiles from Selected Literature Sources

Data source	Impacts (tonnes CO2 equivalent per tonne of textiles recycled)
AEA (2001)	-3.031
ERM (2006 a)	-0.93--1.75
ERM (2006 b)	-7.869
WRATE	-4.290
SCM (2014)	-5.89
SCM (2011)	-14.029
Prognos / IFEU (2008)	-2.18

Notes:
1. Unlike the other studies referenced above, WRAP's values included the benefits associated with avoided residual treatment; these are, however, likely to be minimal for landfilled plastic.
2. Depending on production process and polymer mix

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; ERM (2006 a) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; ERM (2006 b) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; Zero Waste Scotland (2011) The Scottish Carbon Metric, Final Report for Scottish Government, March 2011; Zero Waste Scotland (2012) The Scottish Carbon Metric Carbon Factors, March 2012; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008

The data incorporated within the WRATE model suggests emissions benefits associated with the recycling of donated textiles to be 4.29 tonnes of CO_2 equivalent per tonne of textiles, based on information supplied by Oxfam and WasteSavers. The model assumes that 70% of the clothing donated is not so much recycled, but resold, with 3% being rejected (subsequently landfilled) and a further 27% recycled into rags. The model indicates that impacts are calculated on the basis of a UK-specific mixture of textiles. No further information is provided on the source of emissions reductions data or the composition of textiles. The data contained within WRATE therefore principally relates to material collected through bring banks and charity shops, rather than that obtained through a kerbside collection service – and possibly overestimates the amount of reusable material that might be collected through a kerbside collection scheme.

The earlier version of the Scottish Carbon Metric also assumed a significant proportion of the separately collected textiles were resold, with benefits calculated on the basis of the following assumptions: [14]

- Emissions associated with the production of the total mix of fibres used in textile manufacture are assumed to be 22.3 tonnes CO_2 eq. per tonne of textiles, using data from ecoinvent cited in a study undertaken by BIO Intelligence Service;[15]

14 Detail on assumptions provided through personal communication with Zero Waste Scotland and WRAP
15 Ecoinvent database available from http://www.ecoinvent.ch/

• The metric assumed that 90% of separately collected textiles were suitable for reuse, with the remaining material being suitable for recycling; [16]

• Impacts associated with recycling textiles were assumed to be -0.15 tonne CO_2 equivalent per tonne of textiles (this assumes that the recycled textiles are reprocessed into wipers offsetting low quality paper production);

• 70% of the reused textiles were assumed to offset a new purchase, based on data from Farrant.[17]

Evidence from a number of studies suggests that the reuse rate for material collected through a kerbside system is unlikely to be anywhere close to as high as 70%, let alone 90%. [18]

Equally, however, the above calculations suggest that WRATE has assumed a lower impact for textiles manufacture than is the case elsewhere in the literature.

In the most recent issue of the Scottish Carbon Metric the value is reduced to -5.89 tonnes CO_2 equivalent per tonne of textiles recycled, indicative, perhaps, of a downward revision in the amount of material assumed suitable for reuse. [19]

We have used the data from WRATE in our model. Use of this data implies a relatively conservative reuse rate of around 15% based on the ecoinvent data for textiles manufacturing impacts, assuming that much of the rest of the material could be recycled (and assuming this obtained benefits in line with those attributed in the 2012 version of the Carbon Metric). Although this is likely to understate benefits seen for direct donations to a charity shop, it is probably appropriate for textiles collected through a waste management system, not least given the increased prominence of exchanges through internet sites, as well as charity shops and vintage / 'pre-loved' clothing stores.

Table 2-11: Impacts of WEEE Recycling

Data source	Impacts of WEEE recycling, tonnes CO2 eq. per tonne of product
Small WEEE	-1.482
Mixed WEEE	-1.374
Large WEEE	-1.266

Source: Huisman, J., et al (2008) 2008 Review of Directive 2002/96 on Waste Electrical and Electronic Equipment – Study No. 07010401/2006/442493/ETU/G4, United Nations University, Bonn Germany, cited in Zero Waste Scotland (2011) The Scottish Carbon Metric, report for Scottish Government, March 2011

2.3.7 WEEE

There is also very little data on the impacts of WEEE recycling. WRATE does not consider the benefits of recycling this type of product, and WEEE is not included in either the WRAP review or the earlier AEA study. The Carbon Metric dataset produced for Zero Waste Scotland does, however, include values for WEEE recycling which have been developed from data collated by the United Nations University. These are presented in Table 2 -11. [20]

16 Data provided by textiles reprocessor JMP Wilcox, available from http://www.jmpwilcox.co.uk/products.html
17 Farrant L (2008) Environmental Benefits from Reusing Clothes, Masters Thesis, Technical University of Denmark
18 Nottingham Trent University / Sheffield Hallam University (2008) Public Understanding of Sustainable Clothing, Final Report for Defra, November 2008; MEL (2008) Desktop Textile Waste Study and Compositional Analysis; Report for Oakdene Hollins / Defra, December 2008
19 Zero Waste Scotland (2012) The Scottish Carbon Metric Carbon Factors, March 2012; Zero Waste Scotland (2012) Scotland's Carbon Metric: Keeping you up-to-date with Scotland's Carbon Metric
20 Huisman, J., et al (2008) 2008 Review of Directive 2002/96 on Waste Electrical and Electronic Equipment – Study No. 07010401/2006/442493/ETU/G4, United Nations University, Bonn Germany, cited in Zero Waste Scotland (2011) The Scottish Carbon Metric, report for Scottish Government, March 2011

2.3.8 Values Used in the Model

A summary of assumptions with regard to the climate change impacts excluding biogenic CO_2 emissions – and developed in line with the above discussion - is presented in Table 2 -12.

The majority of studies do not consider the biogenic CO_2 emissions. Data from the UK Environment Agency's tool, WRATE, however, includes these emissions as an information item. We have included the values from this source for paper – which increases the benefit for paper recycling seen in Table 2 -12 by a further 829 kg CO_2 equivalent. Biogenic CO_2 emissions are less important for majority of the dry recyclables other than paper, with the exception of textiles. Since the latter is likely to constitute a mixture of synthetic and natural figures, there will be some biogenic CO_2 impacts; however, this is an issue that has not well covered in the literature to date

Table 2-12: Impacts of Dry Recycling – Values Used in the Model

	Net recycling emissions (excluding biogenic CO2 impacts), kg CO2 eq. per tonne
Paper / card	-315
Plastic	-566
Glass	-201
Textiles	-5,891
Steel	-1,806
Aluminium	-9,985
Food waste	See Section 3
Discarded machines and equipment (includes WEEE)	-181
Others	
Mineral wastes from construction and demolition	2

3. Treatment of Organic Waste

This section presents assumptions included in the model with regard to the treatment of source separated biowaste through composting and AD treatment systems. A review of the literature on this subject is presented in earlier work undertaken by members of the project team for DG Environment, from which many of the assumptions included within the model developed for this project have been taken. [21]

3.1 Data on Composting

Open-air windrow composting processes are those which occur in the open, usually in piles of triangular cross-section, these being turned periodically to introduce air into the process. Alternatively, composting may take place in an enclosed facility – such as in a composting hall. In-vessel composting processes are the commonly used enclosed process. This offers the potential of greater control over some aspects of process management such as a reduction in emissions through the use of abatement technology.

3.1.1 What influences the Emissions Factors

Factors that influence the impact of treating source segregated organic waste include:

- The type of organic waste being treated, key factors being the water content - food waste containing more moisture than garden waste - and carbon content (and within this, the make-up of the organic carbon component, since different types of carbon vary in the extent to which they are amenable to degradation in a given treatment system);
- The type of treatment system used. For AD systems, this also extends to there being different choices for the use of the biogas, which may be used to generate energy through a gas engine or it may be upgraded to produce a relatively pure stream of methane gas. The latter may then be used as a replacement for natural gas in heating systems, or as a replacement for (usually) diesel in powering vehicles;
- Where energy is being generated via the treatment system, the type and source of energy that is assumed to be displaced.
- The avoided emissions from the utilisation of nutrients, and improvement in water retention capacity of soils, resulting from use of soil improving materials, itself affected by the rate at which that displacement takes place.

Methodological factors also have an influence on the results. In particular, the treatment of biogenic CO_2 emissions is a consideration. Recently, authors have recommended the use of a credit to account for un-emitted carbon in landfills and other treatment systems which effectively sequester carbon, where biogenic CO_2 emissions are excluded from the global warming potential calculations in life cycle assessments.[22] This is discussed in more detail in the main report. The issue is of particular relevance for composting systems, where carbon may be stored in the soil following the application of compost, albeit that this storage might not be permanent.

3.1.2 Direct Emissions to Air

Direct emissions to air from the composting of biowaste include impacts resulting from both the composting process itself as well as those associated with compost use.

The quantity of emissions to the atmosphere of any given gas from a given composting process is related to the degree to which the composting process is allowed to proceed towards a theoretical ‘final’ point at which all the carbon dissimilable in the composting process has been degraded.

In practice, different processes may facilitate more or less rapid degradation of the available biomass, so that over a given period of time, different processes may lead to differing levels of emissions. Other things being equal, however, and subject to proper management of the composting process, a longer retention time would be expected to lead to greater ‘raw gas’ (i.e. before the impact of the biofilter) emissions.

21 Eunomia / Arcadis (2010) Assessment of the Options to Improve the Management of Bio-waste in the European Union: Annex F – Environmental Assumptions, Final Report, February 2010

22 Gentil, E., Christensen, T. and Aoustin, E. (2009) Greenhouse Gas Accounting and Waste Management, Waste Management & Research, 27(8), pp696-706; Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., Christensen, T. and Hauschild, M. (2014b) Review of LCA studies of solid waste management systems – Part II: Methodological Guidance for a Better Practice, Waste Management, 34, pp589-606

3.1.2.1 Composting Process

Our assumptions for biogenic CO_2 generation assume the production of a relatively mature compost, such that more of the gas is emitted during the composting phase than would be the case with a less mature product. The remaining non-sequestered carbon is assumed to be emitted during the composting use phase (assumptions for the latter are discussed in Section 3.1.2.2).

There is some debate as to whether methane is emitted in any significant quantities at well managed compost sites. Some have suggested that where process is managed correctly, methane emissions will be negligible as those that occur in the middle of the composting mass will be oxidised at the surface of the composting piles.[23]

For open air facilities we assume emissions to be 50 g of CH_4 per tonne. These values reflect the lowest values seen in Amlinger et al (2008) and are taken to be indicative of well managed composting processes.[24] We also assume emissions of 100 g N_2O per tonne of waste for the same facilities based on the same dataset.

3.1.2.2 Compost Use

Direct emissions associated with compost use include biogenic CO_2 emissions as well as N_2O emissions. In the case of the latter, however, such impacts will also result from the application of synthetic fertiliser. It is therefore important to consider the net impact of using both products.

Some biogenic carbon remains un-emitted following the application to land of compost at the end of the period of analysis. We therefore apply a credit for this sequestration where biogenic CO_2 emissions are excluded from the analysis. The extent to which carbon applied through compost remains sequestered in soil over longer time scales is not known, but most studies indicate that the amount sequestered is reduced over time. In modelling undertaken as part of this report, 5% of the biogenic carbon contained within green waste is assumed to be sequestered through composting at the end of the 100 year period for the analysis. This figure is in line with the median value for these impacts taken from a literature review of a number of LCA studies on composting undertaken in 2009.[25]

More recent work has been undertaken in the United States as part of the Marin Carbon Project, which has looked at the potential for compost application to sequester carbon when applied to grassland. Here the authors have considered in an LCA study – in addition to carbon sequestered as a result of the direct addition to the soil via compost amendment – the potential of organic soil amendments including compost to impact on methane fluxes occurring between the soil and atmosphere.[26] Soil in grassland areas is considered to take up atmospheric methane via the action of methanotrophic bacteria present in the soil. The application of synthetic fertilisers is understood to disrupt this process, whereas organic soil amendments are thought to have less of an impact in this respect.

It is difficult to apply the results of that study to the analysis here, as insufficient detail is available on the assumptions made when undertaking the modelling. In addition, the study considered the impacts for a mixed amendment of manure and compost - the latter only accounted for 25% by mass – and so the results are not entirely comparable to the analysis being undertaken here. Since, however, the total impacts relating to the sequestration impact appear to amount to only 20 kg CO_2 equivalent per tonne of soil amendment, the potential benefit from this additional effect - although interesting - does not appear to be overly significant.[27]

23 Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, Journal of Environmental Engineering, Vol. 130, No. 11, November 1, 2004, p.1394

24 Amlinger F, Peyr S and Cuhls C (2008) Greenhouse Gas Emissions from Composting and Mechanical Biological Treatment, Waste Management and Research, 26, pp47-60

25 Boldrin A, Andersen J, Moller J, Christensen T and Favoino E (2009) Composting and Compost Utilisation: Accounting of Greenhouse Gases and Global Warming Contributions, Waste Management & Research, 27, pp800

26 Delonge M, Ryals R and Silver W (2013) A Lifecycle Model to Evaluate Carbon Sequestration Potential and Greenhouse Gas Dynamics of Managed Grasslands, Ecosystems, 16, pp926-979

27 This figure includes the carbon sequestered via direct application of the compost to soil over a 100 year period. By comparison, our model includes a total credit for carbon sequestration of 31 kg CO2 equivalent per tonne of compost.

Where biogenic CO_2 emissions are included in the analysis, we account for the emissions of the un-sequestered carbon over a 100 year period as indicated above.

3.1.3 Energy Use at Composting Facilities

Assumptions regarding the energy use at composting facilities are presented in Table 3 -13. These are based on data obtained from facilities currently operating in Europe.

Table 3-13: Energy Use at Composting Facilities

	Open Air Facilities	Enclosed (in vessel) facilities
Electricity, kWh per tonne	0	30
Diesel, litres per tonne	1	0

3.1.4 Benefits from the Use of Compost

The application of compost is assumed to displace the requirement for the use of synthetic fertiliser. Avoided impacts are calculated based on the nutrient content of the compost and the impacts associated with the manufacture of fertiliser with an equivalent nutrient content to that of the compost.
Impacts associated with the manufacture of synthetic fertiliser are presented in Table 3 -14, with assumptions used here being taken from the ecoinvent database. Assumptions used with regard to the nutrient content of compost are presented separately in Table 3 -15.
Particularly in the case of the pollution impacts associated with nitrogen-based fertilisers, the literature shows a range of values, as was indicated in a review of the literature undertaken by Boldrin et al in 2009.[28]

Table 3-14: Impacts from the Manufacture of Synthetic Fertiliser

	CO2 eq. emissions per kg of nutrient contained in synthetic fertiliser
Nitrogen (N)	0.007
Phosphorus (P)	0.002
Potassium (K)	0.001

Table 3-15: Nutrient Content of Composts

	Nutrient content (% dry matter content)		
	Nitrogen (N)	Phosphorus (P)	Potassium (K)
Garden waste based compost	1.31%	0.77%	0.70%

Their review suggested climate change impacts per kg of active nitrogen of 4.75 – 13 kg CO_2 equivalent. The equivalent range for phosphorus was 0.52 – 3 kg CO_2 equivalent, whilst that for potassium was 0.38 – 1.53. The ecoinvent data is towards the lower end of the range seen in the literature for nitrogen, and in the middle of the respective ranges for the other two nutrients. As such the ecoinvent data is felt to be reasonably representative of that seen elsewhere in the literature.

28 Boldrin, A., Hartling, K., Laugen, M., and Christensen, T. (2010) Environmental Inventory Modelling of the Use of Compost and Peat in Growth Media Preparation, Resources, Conservation and Recycling, Vol.54, pp.1250–1260

The nitrogen contained in synthetic fertiliser is in a relatively volatile (mineral) form, such that considerable quantities are leached from the soil immediately after application to land. In addition, some of the nitrogen contained within synthetic fertiliser is converted to N_2O following its application to soil.
In contrast to that contained in synthetic fertiliser, the nitrogen in compost is mostly bound up within the organic matter in the product and is much less volatile. As such, less of the nitrogen is available to plants immediately after compost application, as the nutrient only becomes available to plants once it is converted to the mineral – and more volatile - form. Over time, however, the organic nitrogen is gradually converted to the mineral form such that absorption by plants can occur. Some emission of N_2O from the nitrogen contained within compost occurs, but a reduction in such emissions is expected relative to the use of synthetic fertiliser by virtue of the more stable nature of the bound organic nitrogen contained in compost.

The model therefore assumes the slow release of nutrients contained in compost over a 20 year period. It also assumes a reduction in the leaching potential associated with the application of organic nitrogen relative to that of the nitrogen contained within synthetic fertiliser (in the case of the latter, 23% of the nitrogen is assumed to be leached soon after application). The model further assumes a 0.5% reduction in the N_2O emissions from the application of compost (applied in the form of an emissions credit) relative to the case where synthetic fertiliser is used, to account for the reduction in the volatility of the nitrogen contained in compost as described above.[29]

3.2 Anaerobic Digestion

In contrast to composting processes, Anaerobic Digestion (AD) systems degrade the organic waste under anaerobic conditions such that a biogas is produced. The biogas is typically combusted on site generating electricity and heat, although other utilisation routes are used in some member states. In addition to the biogas, the AD process also produces a digestate which is typically applied to land, displacing the use of synthetic fertilisers in a similar manner to that previously described for the composting processes.[30]

3.2.1 Direct Emissions to Air

As is the case with composting processes, direct emissions to air from AD systems result both from the treatment process itself as well as from the use of the digestate. In addition to biogenic CO_2 emissions, some fugitive methane emissions occur. Further emissions impacts arise from the combustion of the biogas during its utilisation for energy generation; as such emissions impacts are typically higher than is the case for composting processes, although the energy generation also results in avoided emissions impacts which are discussed in Section 3.2.2. Assumptions are presented in Table 3 -16.

Table 3-16: Emissions from the AD process

	Emissions impacts, tonnes pollutant per tonne of waste treated
Biogenic CO2 **Food waste** **Garden waste**	 0.45 0.27
CH4 **Food waste** **Garden waste**	0.002 0.001

29 For further discussion of the assumptions used to model these impacts, see Eunomia / Arcadis (2010) Assessment of the Options to Improve the Management of Bio-waste in the European Union: Annex F – Environmental Assumptions, Final Report, February 2010
30 This section discusses the assumptions that have the most influence upon the results of the assessment. A more detailed discussion of the work upon which the model is based (including the remaining assumptions) is available in: Eunomia / Arcadis (2010) Assessment of the Options to Improve the Management of Bio-waste in the European Union: Annex F – Environmental Assumptions, Final Report, February 2010

In addition to the process emissions, additional climate change impacts result from the use of digestate:

• Assumed to be 0.05 tonnes CO_2 equivalent per tonne of feedstock where food waste is the feedstock;

• Assumed to be 0.98 tonnes CO_2 equivalent per tonne of feedstock where garden waste is the feedstock.

3.2.2 Energy Generation and Energy Used by the AD Process

Energy requirements for the AD process are typically met through energy generated at the plant. Benefits from energy generation included within the model account for the use of energy through the AD process, taking into account the electricity and heat used by the AD process.

Biogas generated by the AD process may be utilised in different ways, resulting in different energy generation impacts. Four main options are possible:

• Biogas combustion in a gas engine, from which electricity is exported to the grid – this is the approach most commonly used in European AD facilities;

• Biogas combustion in a gas engine, resulting in the export of electricity as well as the utilisation of heat where suitable outlets for the heat exist;

• Upgrading of the biogas such that bio-methane is produced through the removal of the CO_2 in the gas. The upgraded biogas is then injected into the gas grid (plant utilising the biogas in this way are in operation in Germany, Sweden and the UK);

• Biogas upgrading followed by the use of the bio-methane as a fuel for vehicles (particularly heavy goods vehicles where it displaces diesel). Facilities using this option exist in France and Sweden.

Assumptions regarding the net energy generation for each option are outlined in Table 3 -17, which presents values for food and garden waste. The data confirms that energy generation from garden waste is much lower than that of food waste, as garden waste is more resistant to the anaerobic degradation process.

Table 3-17: Energy Generation from AD Facilities

	Biogas combustion in a gas engine		Upgraded biogas (bio-methane)	
	Electricity (kWh / tonne of waste)	Heat (kWh / tonne of waste)	Gas grid1 (kWh / tonne of waste)	Vehicle fuel2 (litres vehicle fuel / tonne waste)
Food	376	182	915	80
Garden	161	78	395	38

Notes
1. Bio-methane utilised in this way is assumed to offset an equivalent amount of natural gas.
2. Bio-methane utilised in this way is assumed to offset an equivalent amount of diesel combusted in a heavy goods vehicle

3.2.3 Benefits from the Use of Digestate

3.2.3.1 Air Emissions Impacts Avoided through Displacement of Synthetic Fertiliser

Digestate is assumed to displace the use of synthetic fertiliser in a similar manner to that previously described for compost. Assumptions for the nutrient content of digestate are presented in Table 3 -18. [31] The data from this table is combined with the information previously presented in Table 3 -14 - which provides assumptions on the pollution impacts per

[31] Data provided through personal communication from WRAP (collated from a series of field studies undertaken in the UK)

Table 3-18: Nutrient Content of Digestate

	Nutrient content (kg per tonne of fresh weight of digestate)		
	Nitrogen (N)	Phosphorus (P)	Potassium (K)
Food waste	4.78	0.4	2.3
Garden waste	1.9	0.2	0.5

tonne of synthetic fertiliser – to calculate the avoided pollution impacts from the use of digestate in place of synthetic fertiliser.

3.2.3.2 Indicative Emissions Factors

Indicative emissions factors are shown in Table 3 -19. Data is derived from the European Waste Model developed by Eunomia. For electricity only generation, the type of fuel used for the electricity that would otherwise have been generated using other sources has an impact; the table therefore shows impacts with gas, coal and wind generation. These effects are discussed further in the subsequent sections on incineration. For vehicles, it is assumed that diesel is displaced by the use of biogas.

Many facilities generate only electricity. The table shows that benefits are significantly reduced where the avoided source of electricity generation is energy generated using wind turbines. As was indicated previously, in many Member States the proportion of renewable (or low carbon) energy is expected to increase in the future, reducing the benefits seen here for AD with electricity or CHP. In contrast, the decarbonisation of the transport system is much less well advanced, and so the benefits seen here for AD where the upgraded biogas is used to fuel vehicles are expected to remain at this level for many years to come.

Table 3-19: Impacts of Source-Segregated Organic Waste Treatment

	Garden waste, kg CO2 eq. per tonne waste	Food waste, kg CO2 eq. per tonne waste
Windrow Composting	21	29
In-vessel Composting	41	49
AD – electricity only (gas avoided)	-120	-150
AD – electricity only (coal avoided)	-223	-331
AD – electricity only (wind avoided)	-66	-63
AD – CHP	-137	-185
AD – Upgraded biogas used in gas grid	-143	-195
AD – Upgraded biogas fuelling vehicles	-180	-280

4. Treatment of Residual Waste

4.1 Residual Waste Composition

When modelling the performance of the residual waste treatment systems – landfill, incineration and MBT – an understanding of the residual waste composition is required. This is likely to be affected by the level of recycling taking place in the country, as well as the consumption behaviour of inhabitants; this, in turn, is related to a certain extent to income. The data may also affected by the extent to which commercial waste is collected by authorities. Examples of residual waste composition data from a number of European countries are presented in Table 4 -20. The data was derived from information obtained during the development of the European waste model. There is some variation in the composition data from the different countries; for example, Malta has a very high level of food waste in the residual waste stream, in part as a consequence of the collection of wastes from the hospitality sector (e.g. hotels, restaurants) alongside the household waste stream. The proportion of paper, glass, and plastics is higher in Bulgaria than in the UK as levels of separate collection are higher in the UK than is the case for Bulgaria

Table 4-20: Example Residual Waste Composition Data

	NL	MT	UK	BG	Used
Food	33%	52%	38%	29%	36%
Garden	8%	0%	3%	7%	5%
Wood	3%	0%	1%	1%	1%
Paper / Cardboard	17%	18%	14%	21%	17%
Textiles	4%	2%	4%	3%	3%
Glass	5%	6%	4%	10%	6%
Steel cans	3%	3%	2%	1%	2%
Aluminium cans	1%	1%	1%	1%	1%
Plastics	13%	12%	15%	12%	13%
WEEE	1%	0%	1%	0%	1%
Rubble, soil	4%	0%	1%	3%	2%
Inert	0%	3%	4%	7%	4%
Miscellaneous combustible material	7%	3%	11%	6%	7%

Source: Eunomia European Waste Model

The residual composition data was used to model the impact associated with treating one tonne of residual waste by each of the waste treatment methods considered in the analysis. A “typical” residual waste composition was developed using a weighted average of the four sets of composition data.

4.2 Landfill

4.2.1 What influences the Emissions Factors

Emissions factors for landfilled waste are influenced by:

• The composition of material landfilled:

o Only organic materials such as food waste and paper degrade in landfill, materials containing fossil carbon such as plastics do not degrade, and neither do inert materials such as metals and glass;

o The type of carbon contained within the material is also important, as well as the amount. The carbon contained in woody garden waste materials and some types of paper degrades very slowly in landfill, whilst that contained in food waste is much more readily degradable;

• Climatic factors: degradation rates are faster in damp climates, whilst the rate also increases with temperature;

• Landfill gas management, particularly the capture of landfill gas for energy generation and flaring, as well as the way in which the captured gas is used (usually for electricity generation in CHP units).

Although for the majority of other parts of the waste management system the main greenhouse gas emitted is carbon dioxide, in landfill, the principle greenhouse gas produced is methane. Unlike some other treatment systems such as incineration, where impacts are more or less instantaneous at the point of treatment, emissions from landfilling continue to occur over a considerable time period.

It will be seen in this section that the IPCC provides guidance on the modelling of landfill emissions. As with other elements of the waste management system, methodological factors are important in determining the outcome. Landfill gas modelling, in particular, is subject to considerable uncertainty; assumptions in several key areas vary in the literature, and this, in turn, affects choices used in country inventories, a key source of information on landfill impacts.

The treatment of the biogenic CO_2 emissions is an important consideration given that – according to the IPCC methodology – over 50% of the biogenic carbon is not expected to appreciably degrade over the 100 year time period over which emissions assessments are typically considered. The application of the credit for the sequestered carbon is therefore particularly relevant where landfill impacts are concerned.

4.2.2 The GWP of Methane

The publication of the Fifth Assessment Report by the IPCC saw it update the global warming potential (GWP) values for N_2O and CH_4 from those presented in the Fourth Assessment Report. The various GHGs differ in the capacity for capturing and re-radiating outgoing infrared radiation, and as such, the contribution made to radiative forcing, which is the basis for the GWP values. These values, in turn, are used in life-cycle assessment studies to calculate the climate change impacts of products and systems in terms of carbon dioxide equivalent emissions. Values from several IPCC assessment reports are presented in Table 4 -21, which shows the 100-year GWPs.[32]

Table 4-21: 100 year GWPs published by the IPCC

			Fifth Assessment Report	
GHG	**Second Assessment Report**	**Fourth Assessment Report**	**Without climate carbon feedback**	**With climate carbon feedbac**
CO2	1	1	1	1
CH4	21	25	28 / 301	34
N2O	310	298	265	298
Notes: The lower value is for biogenic methane, the higher one fossil-methane				

Sources: IPCC (1995) IPCC Second Assessment Report. A Report Of The Intergovernmental Panel On Climate Change; IPCC (2007) IPCC Fourth Assessment Report: Climate Change 2007. Working Group I: The Physical Science Basis; IPCC (2013). Climate Change 2013: The Physical Science Basis. Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Chapter 8: Anthropogenic and Natural Radiative Forcing

32 The radiative forcing effect varies over time and thus the IPCC publishes figures for 20, 100 and 500 year GWPs, with the 100-year value being the one that is the most widely used in policy analysis and life cycle assessments

The table confirms that the values for CH_4 have been increased over the years, whilst in contrast, those for N_2O have been decreased. For the first time, two values for CH_4 are published in the Fifth Assessment Report, showing the impacts both including and excluding climate carbon feedbacks. It is noted in the report that the inclusion of feedback increases the uncertainty associated with the results, but that the inclusion also ensure greater methodological consistency. The IPCC does not, however, provide an opinion as to which of the two values should be used.

A recent report confirmed that a number of organisations have differed in their approach to adopting the new values.[33] Adoption of the new values has been relatively slow, as national and international agencies emphasise year-on-year consistency between inventories. In line with this, communication with DG CLIMA made in the context of developing Eunomia's European Waste Model confirmed that the updated values would not be adopted into European climate change policy until 2020 when the next set of emissions reduction targets are due to be agreed. Where the updated values have been adopted, agencies have shown a preference for the values without the climate feedback. France's Agence de l'Environnement et de la Maîtrise de l'Énergie (ADEME), now applies the GWPs without feedback in the latest update to its "Base Carbone" tool.[34] The EPA's 2014 U.S. GHG Inventory also notes that AR5 GWP values without feedback have a calculation methodology that is more consistent with those that were used in the AR4 report.[35]

The above discussion is focussed on the 100 year GWP as this is the value which is most widely used in analyses of this nature. However, other authors have confirmed that the choice of time horizon is a subjective one which cannot be justified on scientific grounds.[36] Over the shorter time horizons the GWP of methane rises, such that the value for the 20 year GWP is 72.

4.2.3 Landfill Gas Generation

4.2.3.1 Variation in Member State Models

IPCC methodology uses first order decay equations to estimate landfill gas generation. There are many models considering this in the literature; the majority use a similar approach to that of the IPCC. An examination of these models therefore appears to be a good starting point when considering landfill gas generation modelling.

Countries can use their own assumptions for many of the factors in the model, which is intended to reflect the situation for the country. A review of these models confirms the sources of variation between the models to be the following:

- Climatic factors;
- Assumptions about the rate of decay occurring for each of the different fractions;
- Variation in fraction of carbon that forms methane;
- The proportion of carbon that is assumed to be dissimilable;
- Which month the decay is anticipated to start in.

In the case of the first of these factors, the IPCC methodology confirms that in wet climates the amount of methane generation should be higher and that in much hotter temperatures it will be greater. However IPCC makes a distinction only between tropical and boreal / temperate climates so all European countries will be in the first category. In Europe, then, it mostly comes down to variation in moisture. We would expect to see variation across different countries relating to this, and indeed some differentiation is seen.

Eunomia's European Waste Model is based on, as far as landfill is concerned, data from each country's IPCC model. Taking food waste as an example:

- Spain's model (in climate zone 1) assumes 0.033 tonnes of methane is generated per tonne of food waste;
- The UK model (in climate zone 2) suggests 0.040 tonnes of methane for the same amount and type of waste;

33 Econmetrica (2015) Understanding the Changes to Global Warming Potential (GWP) Values, February 2015

34 ADEME (2014) Base Carbone : Les gaz

35 EPA (2014) U.S. Greenhouse Gas Emissions and Sinks: 1990-2012. Annex 6 Additional Information

36 Brandao, M., and Levasseur, A. (2010) Assessing Temporary Carbon Storage in Life Cycle Assessment and Carbon Footprinting: Outcomes of an Expert Workshop, JRC Scientific and Technical Reports, JRC European Commission, Brussels

• The model of the Czech Republic (also in climate zone 2) yields 0.059 tonnes.

When assumptions between the second and third of these are compared, the following can be noted:

• The Czech model assumes 61% of the carbon forms methane whereas the UK has 50%;

• The proportion of dissimilable carbon is 60% whereas the UK assumes 50%,

• There is a variation in the delay time.

This is something of an extreme example since the Czech Republic model yields the highest methane generation results when compared to all of the others for this type of material, but it is noted that the values for Ireland and Italy are similar, despite being in different climate zones.

4.2.3.2 Decay rates

A key factor in these models is the decay constant, k. Estimating the values of the decay constant in real landfill conditions is difficult. The study by Oonk in 1994 (also part of a 1995 measurement report) and a similar exercise in USA in more arid conditions (by Gregg Vogt) appear to be the only field-studies that have been performed that shed light upon values for k.[37] The study by Oonk estimated values of 0.1 for 'mixed waste', or 0.185, 0,1 and 0.03 for fast, moderate and slowly degrading waste, respectively, when using a multi-phase (i.e. a model with more than one decay constant) model. This appears to be the only information that comes from actual field-data, and it is the data upon which the IPCC default values are based.

4.2.3.3 Methane Content of Landfill Gas

The assertion that 50% of landfill gas by volume is methane is widely held, and appears to be the default assumption in the five models considered in a Canadian model calibration study.[38] However, Afvalzorg assume 56% of the gas to be CH_4, whilst Oonk, in a literature review on CH_4 generation from landfills, indicated a range of possible CH_4 concentrations in landfill gas of between 45 and 60% - the latter echoing the range of values proposed in earlier analysis by Tchobanoglous et al.[39]

Methane formation only occurs in moist, airless spaces.[40] Whilst an increase in the moisture content leads to more methane formation, the presence of oxygen, on the other hand, prevents methane from forming. The proportion of methane in landfill gas is thus dependent upon the percentage of moisture and the absence of oxygen at any given time. In a situation where conditions are sub-optimal for methanogenesis, the resulting CH_4 fraction of the landfill gas may be as low as 35%. Alongside this, the composition of landfill gas is likely to vary over the life of landfill, as a consequence of both the stages of methanogenesis and landfill gas management practices.

4.2.3.4 Starting Point for Decay

During both the early stages of degradation (such as the acidogenesis phase) and the early part of the methanogenesis phase, the gas is likely to have a greater proportion of CO_2 in comparison to its methane content.[41] Once the main phase of methanogenesis is underway, the concentration of methane generally increases relative to that of CO_2 – influenced in part by the stoichiometry. As methanogenesis slows the concentration of CO_2 again rises relative to the amount of methane.

Landfill management practices are also likely to influence the relative proportions of the two gases over time. Once applied, the permanent cover of the landfill acts as a barrier to moisture, reducing CH_4

37 Oonk H, Weenk A, Coops O and Luning L (1994) Validation of Landfill Gas Formation Models, Dutch Organisation for Applied Scientific Research, Report no 94-315; Oonk H and Boom T (1995) Landfill Gas Formation, Recovery and Emission, TNO-rapport 95-203,; Vogt G., Augenstein D., (1997): Comparison of models for predicting landfill methane recovery, SCS Engineers, Report File No. 0295028, Reston, Virginia, USA

38 Thompson S, Sawyer J, Bonam R and Valdivia JE (2009) Building a better methane generation model: Validation models with methane recovery rates from 35 Canadian landfills, Waste Management, 29, pp2085-2091

39 Oonkay (2010) Literature Review: Methane from Landfills: Methods to Quantify Generation, Oxidation and Emission; Jacobs J and Scharff H (u.d.) Comparison of Methane Emission Models and Methane Emission Measurements, NV Afvalzorg, The Netherlands; Tchobanoglous G, Hilary T and Vigil S (1993) Integrated Solid Waste Management: Integrated Principles and Management Issues, McGraw-Hill, New York

40 Center for a Competitive Waste Industry (2008) Landfill Gas to Energy Compared to Flaring

41 Tchobanoglous G, Hilary T and Vigil S (1993) Integrated Solid Waste Management: Integrated Principles and Management Issues, McGraw-Hill, New York

formation. At the same time, however, this will also reduce the availability of leachate within which the remaining CO_2 can dissolve.

Delay in the starting of the decay process. Process is described in recent work by Golder Associates for Defra in the UK:[42]

In stage 1 of landfill gas generation, waste degrades aerobically, like compost, consuming the air which surrounds it. Only when this air has been consumed does Stage 2 commence, which is the start of acidogenic waste degradation. This is characterised by carbon dioxide and hydrogen generation, and no methane is produced at this stage. Waste is hydrolysed and degrades to produce long chain organic acids. Stage 3 is known as the acetogenic phase, when carbon dioxide and hydrogen production peaks, methane is starting to be generated, and acetic acid is a degradation product. Landfill gas generation reaches its peak in stage 4, the fully methanogenic phase.
…. The start of stage 4, which is fully anaerobic methane production, has since been demonstrated in the UK by Barry et al (2004) with methane production achieving recovery rates in the sixth month after placement in a new waste cell. This means that stages 1-3 occur in fresh waste over a typically six month timeline in a landfill's 100 year plus gas generation lifetime. This could be represented in a model as a six-monthly delay in gas generation from the time of emplacement.

An earlier review confirmed that the delay in the commencement of methanogenesis could be for up to a year, and that this might vary depending on climatic factors and landfill management processes.[43] The impact could be potentially significant as relatively little landfill gas is captured in the first few years of operation in most landfills since the permanent cap is not installed until after this point. In practice, however, most models do not account for the variation in capture rate over time – using a fixed rate over the lifetime of the landfill.

4.2.3.5 Proportion of Dissimilable and Degradable Carbon

The IPCC default factors make clear that a significant proportion of the carbon within the waste is not degraded appreciably over a 100 year period. In effect, therefore, a substantial amount of biogenic material remains sequestered in the landfill.It is for this reason that there is a need to account correctly for the biogenic CO_2 emissions that do actually result from landfill during this period. This is discussed in more detail in the main report.

4.2.4 Gas Collection and Management

The wider literature suggests a range of estimates for the efficiency of gas collection with a distinction being made between instantaneous collection efficiencies and the proportion of gas that can be captured over the lifetime of the landfill.[44] Whilst instantaneous collection rates for permanently capped landfilled waste can be as high as 90%, capture rates may be much lower during the operating phase of the landfill or when the waste is capped with a temporary cover.[45]
In addition, gas collection is technologically impractical towards the end of the site's life. The Intergovernmental Panel on Climate Change (IPCC) has recently stated that lifetime gas capture rates may be as low as 20%.[46] A previous study by the European Environment Agency uses the IPCC figure.[47] A review of the literature undertaken in 2011 in this respect is presented in Table 4 -22, and confirms the considerable range in estimates of collection efficiencies from the different sources.

44 Anderson P (2005) The Landfill Gas Recovery Hoax, Abstract for 2005 National Green Power Marketing Conference; USEPA (2004) Direct Emissions from Municipal Solid Waste Landfilling, Climate Leaders Greenhouse Gas Inventory Protocol – Core Module Guidance, October 2004; Brown K A, Smith A, Burnley S J, Campbell D J V, King K and Milton M J T (1999) Methane Emissions from UK Landfills, Report for the UK Department of the Environment, Transport and the Regions
45 Spokas K, Bogner J, Chanton J P, Morcet M, Aran C, Graff C, Moreau-Le Golvan Y and Hebe I (2006) Methane Mass Balance at 3 Landfill Sites: What is the Efficiency of Capture by Gas Collection Systems? Waste Management, 5, pp515-525
46 IPCC (2007) Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (Metz B, Davidson O R, Bosch PR, Dave R, and Meyer L A (eds)), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., pp 600
47 Skovgaard M, Hedal N, Villanueva A, Andersen F and Larsen H (2008) Municipal Waste Management and Greenhouse Gases, ETC/RWM Working Paper 2008/1, January 2008

Table 4-22: Overview of Collection Efficiencies from the Literature

Landfill type	Reference	Method	Gas extraction efficiency	Remarks
Partial ex-ploit-ation	Oonk (1994)	Engineering consi-derations	20%	Based on knowledge of experienced engineers
Mostly closed	Oonk (1995)	MBM-measure-ments of CO2 and CH4	11-52%	3 Dutch landfills
	Ehrig (1999)	Engineering consi-derations, validated by comparing extractions and models	40-60%	German landfills in exploi-tation and closed landfills. Validation suggests effi-ciency is overestimated
	Mosher (1999)	Static chamber and tracer plume measurements of methane	70%	One USA landfill, partly in operation, partly sealed with a geo-membrane. Me-thane oxidation assump-tions unclear
	Scharff (2003)	MBM-measure-ments of CO2 and CH4	10-55%	4 Dutch landfills
	Michaels (2006)	Gas extraction com-pared to prognosis	75-85%	Wisconsin landfills, ef-ficiency dependent on assumed model for LFG generation
			46-54%	
	Lohila (2007)	Micrometeorologi-cal method	69-78%	Emission reduction upon start-up of collection at Finnish landfill. Note the applicability of the mea-surement method is cur-rently under discussion.
	Themelis (2007)	Gas extraction com-pared to prognosis	35%	Average value taken across 25 Californian landfills. As-sumptions for gas genera-tion are very uncertain.
	Borjesson (2007)	CH4 emission and oxidation measure-ments	33-64%	4 Swedish landfills
	Oonk (2010)	Gas extraction com-pared to prognosis	15% 45%	State of the art & non-state of the art Dutch landfills

Recently closed	Oonk (1994)	Engineering considerations	45-60%	Based on knowledge of experienced engineers
	Oonk (1995)	MBM-measurements of CO2 and CH4	10-80%	9 Dutch landfills, sand cover
	Spokas (2005)	CH4 emission and oxidation measurements	88-92%	1 French landfill, 30cm clay cover
	Borjesson (2007)	CH4 emission and oxidation measurements	14-65%	2 Swedish landfills
Less recently closed	Oonk (1994)	Engineering considerations	60-95%	Based on knowledge of experienced engineers
	Oonk (1995)	MBM-measurements of CO2 and CH4	96-100%	2 Dutch landfills, clay and geo-textile cover
	Mosher (1999)	Static chamber and tracer plume measurements of CH4	90%	1 USA landfill. Result somewhat unreliable due to inaccuracies in measured extraction
	Spokas (2005)	CH4 emission and oxidation measurements	84-93%	3 French landfills, clay and geo-textile caps
	Spokas (2005)	CH4 emission and oxidation measurements	40%	1 French landfills, geo-synthetic clay
	Huitric (2006)	CH4 emissions	93-96%	1 Californian landfill 1.5m clay
	Huitric (2007)	CH4 emissions	99%	Same Californian landfill 5 years later

Source: Eunomia Research & Consulting (2011) Inventory Improvement Project - UK Landfill Methane Emissions Model, Report for DEFRA, January 2011

Reflecting the uncertainties associated with understanding these impacts, countries have historically reported different values for the proportion of gas captured within the inventory submission made to the UN-FCCC. Table 4 -23 presents the methane capture efficiency for selected countries as reported in 2008. In general, those monitoring the impacts have reported much lower capture efficiencies than those where the capture efficiency is estimated.

The table shows that the capture efficiency previously reported by the UK is relatively high in comparison to that reported by other countries. The UK has previously justified this value with reference to the efficacy of the gas collection systems employed within UK landfills which start to recover gas even during the filling stage before the landfill cell has been capped. Recent work by Golder Associates to validate this figure has, however, suggested a 52% lifetime recovery rate even for the UK landfills, also suggesting an instantaneous capture rate of 68%.[48]

48 Golder Associates (2014) Review of Landfill Methane Emissions Modelling, Report for Defra

Table 4-23: Methane Emissions and Recovery Reported to UN-FCCC and Calculated National Recovery Efficiency in 2008

		Emission (Gg)	Recovery (Gg)	Methane capture efficiency
Austria	monitored	74	15	15%
Denmark	monitored	15	5	8%
Germany	estimated	358	526	57%
The Nether-lands	monitored	233	44	15%
UK	estimated	960	2,561	71%
USA	estimated	6,016	6,451	49%

Source: Data retrieved from CRF's of individual countries, to be found at: http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/5270.php

Reflecting this work Defra used the 60% in its carbon modelling study as the default capture rate.[49]
Reflecting these issues, Eunomia's European Waste Model applied a default capture rate of 50% to all the IPCC gas generation models. It was recognised that older sites may perform worse than this; however, the aim was to consider the performance of landfills that are still accepting waste, and which are therefore expected to be compliant with the Landfill Directive.

4.2.5 Values used in the model

Gas generation data used in the analysis here has been derived from the UK's IPCC model, as the data from this was relatively close to the average across the European countries. Three capture rates were used in the analysis (20% / 50% / 70%), to show the variation in performance from changing this performance characteristic. As was previously discussed, the figures are also calculated including a credit to account for the carbon that is un-emitted after 100 years where biogenic CO_2 emissions are excluded from the analysis.

4.3 Incineration and Gasification

Incineration involves the generation of energy through combustion, whilst in gasification the energy is generated through chemical conversion step - leading to greater flexibility with regards to the energy outputs from the process. Whilst incineration processes are reliant on the generation of electricity through a steam turbine (where generation efficiencies are somewhat limited) gasification processes have, in theory, the potential to utilise energy generation technologies that can operate at higher generation efficiencies.
The climate change impacts of incineration and gasification are influenced by:

- The carbon content of incinerated materials, along with the type of carbon. Typically only the fossil carbon is included in emissions where a life cycle assessment approach is taken. Waste that is high in plastic will generate more energy, but this will also result in significant quantities of fossil CO_2 being released;
- The amount and type of energy generation (electricity or heat): here the literature indicates a considerable variation in performance of facilities;
- The type of energy source that is displaced by energy generation at the incinerator – when electricity is generated, for example, greatest benefits are seen where coal is displaced, whilst the displacement of generation from nuclear or solar would result in little or no benefit. Efforts made by countries in future years to decarbonise the electricity supply will therefore result in a progressive decline in the

49 Defra (2014) Energy Recovery for Residual Waste: A Carbon Based Modelling Approach

benefits seen from electricity generation from waste facilities;

• The extent to which removal of materials for recycling: metals are typically recycled from bottom ash, leading to the benefits described in Section 2.2;

• To a lesser extent, the amount (and type) of energy used in the process also has an impact.

Although there is the potential for gasification processes to achieve efficiencies that are higher than incineration facilities, in practice the vast majority of gasification plant currently utilise the same energy generation technology as incinerators, i.e., the steam turbine. Given that the chemical conversion step in gasification itself results in some energy losses, these facilities therefore typically achieve a lower efficiency than is seen at incineration plant.

4.3.1 Generation efficiency

4.3.1.1 Incineration

The literature suggests a wide range of performance in respect of energy generation performance. Information received by Eunomia from the Confederation of European Waste-to-Energy Plants (CEWEP) when developing the European Waste Model suggested typical net generation efficiencies for European plant to be 17% net generation for facilities generating only electricity, efficiencies of 14% electricity with 41% heat for CHP plant, and 70% efficiency for those generating heat only.[50] A number of papers, looking at the performance of incineration facilities generating only electricity in the context of life cycle assessment, have assumed efficiencies in line with those of the CEWEP.[51]

On the other hand, planning documents confirm that many proposed facilities in the UK – also generating solely electricity - are expected to achieve higher net electrical generation efficiencies, suggesting net electrical generation efficiencies of between 22-28%.[52]

The literature also suggests that facilities operating in CHP in some countries have far better generation efficiencies; one recent paper looking at the situation in Denmark suggests an upper end performance of 21% electricity with 74% heat, whilst another paper gave a total generation efficiency for a plant in Sweden of 108% (assuming 19% electricity with 81% heat).[53] In both these cases, although not clearly stated in either paper, it is unlikely that the figures for electricity represent the electrical generation efficiency, as is presented above for the electricity-only facilities. Rather, this figure is likely to be the proportion of thermal energy used to generate electricity.[54] Electrical generation efficiencies for the CHP plant are likely to be in the order of 7-8% when the conversion from thermal to electrical energy is taken into account.

Other feedback received during the consultation authorities undertaken during the development of Eunomia's European waste model suggested the following efficiencies for the better performing CHP plant in Europe (noting that these were suggested as future-looking "typical" efficiencies):

• Sweden: 12% electricity together with 83% heat;

• Finland: 23% electricity and 65% heat.

Again it seems likely this is the percentage of thermal energy being used to generate electricity, rather than the actual electrical generation efficiency.

50 Email confirmation received from Ella Stengler of the CEWEP

51 Assamoi, B. and Lawryshyn, Y. (2012) The Environmental Comparison of landfilling vs incineration of MSW accounting for waste diversion, Waste Management, 32(5), pp1019-1030; Belboom, S., Digneffe, J. M., Renzoni, R., Germain, A. and Leonard, A. (2013) Comparing Technologies for Municipal Solid Waste Management using Life Cycle Assessment Methodology: a Belgian Case Study, Int. J. LCA, 18(8), pp1513-1523; Zaman, A. (2010) Comparative study of municipal solid waste treatment technologies using life cycle assessment method, Int J Environ. Sci. Tech., 7(2), pp225-234

52 Hitachi (u.d.) Ferrybridge Multi-fuel Plant / UK: Energy from Waste Plant, Hitachi Zosen Inova AG, Switzerland; Veolia Environmental Services (2011) Hertfordshire County Council – ISFT – A WRATE Assessment of the VES (UK) Proposed Solution, Veolia United Kingdom, London; Cory Environmental, Wheelaborator Inc. and RPS (2012) Willows Power & Recycling Centre Environmental Permit Application: Main Permit Application; Fichtner (2012) AmeyCespa: City of York and North Yorkshire PFI WRATE Model, AmeyCespa, Cambridge

53 Merrild H, Larsen A and Christensen T (2012) Assessing Recycling Versus Incineration of Key Materials in Municipal Waste: the Importance of Efficient Energy Recovery and Transport Distances, Waste Management, 32, pp1009-1018; Bernstad A, Jansen J and Aspegren (2011) Life-cycle Assessment of a Household Solid Waste Source Separation Programme: A Swedish Case Study, Waste Manag Res, 29, pp1027-1042

54 The impact of this can be seen in the methodology for the R1 calculations. Electricity generation in (GJ) is weighted by 2.

4.3.1.2 Gasification

Energy generation in the gasification process occurs in an environment that is relatively low in oxygen. Under such circumstances, combustion cannot occur. Instead, when heat is applied in the gasifier, the waste goes through a chemical conversion process producing “synthesis” gas or syngas, and it is this syngas that contains the energy from the process. The syngas can then be used to generate steam in the boiler, and this in turn used to generate electrical energy in the same way as is done in the incineration process.

There are relatively few facilities utilising gasification to treat municipal solid waste in comparison to those using combustion techniques, although the technique is fairly well utilised to treat other feedstocks including coal and biomass. A database produced by the Gasification Technologies Council identified only 23 facilities currently in operation globally.[55]

Where this approach is used, the most widely utilised approach to energy generation is electricity generation using a steam turbine or boiler - as this poses the least technical challenges. Numbers of plant using a gas engine to generate electricity following the gasification of MSW are fewer still. A separate review of the technology undertaken in 2008 indicated there were four plant in operation at commercial scale using such an approach, although no details were provided.[56] The Thermoselect technology uses this approach, and there are four example facilities in operation in Japan, whilst other analysis has identified several other facilities worldwide (however, at least one of these has since closed).[57]

In recent years, several operators have developed plasma gasification technologies. A review of the global take-up of thermal gasification technology published in 2013 confirmed that at the time of writing only, Westinghouse technology (Alter-NRG) had plant operating at a commercial scale for this type of technology, this being the Utashinai plant operated by Hitatchi metals in Japan.[58] A detailed review of the technology indicated that this plant did not operate solely on MSW (it also treated tyre residues) and that it generated energy using steam turbine technology rather than a gas engine.[59] Plasma gasification has been used by Westinghouse to treat other feedstocks; there is a relatively small plant treating hazardous waste in India (at Pune), and a biomass facility in China. At the time of writing however, there do not appear to be any facilities using a gas engine in combination with plasma gasification, operating at commercial scale and treating MSW.[60]

The chemical conversion process typically results in greater energy losses in comparison to the incineration process. As such, where energy generation is undertaken using steam turbine technology, overall efficiencies will be less than those seen for incineration plant. This is reflected in the literature; a recent review paper indicated a range of electrical generation efficiencies for gasification plant of 15-24%.[61]

4.3.2 Source of Displaced Energy

One of the key assumptions in this type of analysis is the carbon intensity of electricity generation. The average mix of fuels used to generate power in the electricity grid taken at a particular point in time is commonly used where the carbon footprint of an individual facility is concerned. Where, however, the consequences of a decision are being modelled – as is the case where the development of a new facility is concerned - a number of authors have indicated it is appropriate to use marginal energy data in waste management LCA. [62] More generally, marginal data reflects the consequences of small changes in the quantity produced of a good or service. Where as-

55 Available from http://www.gasification.org/

56 Juniper (2008) Independent Waste Technology Report: The Alter NRG / Westinghouse Plasma Gasification Process

57 Waste Catalog project, Juniper (2008) Independent Waste Technology Report: The Alter NRG / Westinghouse Plasma Gasification Process

58 Fabry F, Rehmet C, Vandad Rohani and Fulcheri L (2013) Waste Gasification by Thermal Plasma: A Review, Waste Biomass Valor, 4, pp421-439

59 Juniper (2008) Independent Waste Technology Report: The Alter NRG / Westinghouse Plasma Gasification Process

60 Westinghouse Plasma Corporation (2014) Westinghouse Plasma Gasification: Scaling up to 100 MW, SGC International Conference on Gasification, Malmo, Sweden

61 Arena U (2012) Process and Technological Aspects of Muncipal Solid Waste Gasification, A Review; Waste Management, 21, pp625-639

62 Ekvall.T, and Weidema, B.P. (2004) System Boundaries and Input Data in Consequential Life Cycle Inventory Analysis, International Journal of LCA, Vol.9, No.3, pp.161–171; Gentil, E., Christensen, T. and Aoustin, E. (2009) Greenhouse Gas Accounting and Waste Management, Waste Management & Research, 27(8), pp696-706

sumptions for the marginal generation source in LCA are concerned, this principally relates to the estimate of the next generation facility to be built, given economic, political and resource constraints.
Data on the carbon intensity of key electricity generation sources is presented in Table 4 -24. This confirms that impacts for renewables and nuclear are considerably than that of coal; gas CCGT is in the middle of these two extremes.

Table 4-24: Carbon Intensity of Key Electricity Generation Sources

	Emissions g CO2 per kWh electricity	Sources
Gas CCGT	360 - 575	EIB, Weisser
Coal	800 – 1,000	ecoinvent
Nuclear	mars-24	Weisser
Renewables (excluding biomass)	janv-50	Weisser

Sources: EIB (2014) European Investment Bank Induced GHG Footprint – Methodologies for the Assessment of Project GHG Emissions and Emission Variations Version 10.1, April 2014; ecoinvent database; Weisser D (2007) A Guide to Life-cycle Greenhouse Gas (GHG) Emissions from Electric Supply Technologies, Energy, 32, pp1453-1559

The UK has published figures on both the current and future long run marginal source of electricity generation which are required to be used in policy appraisal.[63] The figures are updated annually, and the data shows a decline in the carbon intensity of the marginal source of energy which can be assumed to be avoided by those new projects that are not themselves so large that they influence the marginal carbon intensity which is in line with the emissions reduction targets contained within the Climate Change Act. This data was used in the recent analysis undertaken by Defra on the relative performance of incineration and landfill.[64] The carbon intensity for the current marginal is close to that of gas generation using Combined Cycle Gas Turbine (CCGT) assuming the carbon intensity of the latter to be similar to that indicated by Defra.

The situation is somewhat less clear in some other countries, which have yet to publish trajectories of emissions reductions relating to power plant. In Germany, for example, relatively little information exists in respect of marginal sources of generation – a relatively recent report on German biomass energy generation used marginal data from 2007. [65] In recent years, a significant proportion new coal generation capacity has been developed in Germany, and a number of substantially sized coal facilities are due to become operational over the next few years. [66] This data suggests the current marginal fuel for electricity generation in Germany to be coal.

However, the situation is expected to change in Germany in the future, with a reduction in the construction of new coal-fired generation being forecast, and increasing renewable capacity being developed. Other sources confirm the country has ambitious renewable electricity generation targets of 35% by 2020 and 50% by 2030.[67] Although coal is expected to continue to be a significant contributor to the mix of fuels used for electricity generation in the future, one review of policy has confirmed the country also has ambitious plans for the development of carbon

63 DECC and HM Treasury (2013) Appraisal Guidance: Energy Use and GHG Emissions: Supporting tables 1-20, Supporting the Toolkit and the Guidance, HM Treasury, London
64 Defra (2014b) Energy Recovery for Residual Waste: A Carbon Based Modelling Approach, Defra, London
65 Buhle L, Stulpnagel R and Wachendort M (2011) Comparative life cycle assessment of the integrated generation of solid fuel and biogas from biomass (IFBB) and whole crop digestion (WCD) in Germany, Biomass and Bioenergy, Vol 35(1), pp363-373
66 Poyry (2013) Outlook for New Coal-fired Power Stations in Germany, the Netherlands and Spain, Report to DECC
67 http://www.iea.org/media/training/bangkoknov13/session_4b_germany_generation.pdf

capture and storage technology for its coal plant in the future.[68] This indicates a similar trajectory for emissions reduction is in place for electricity generation in Germany as is suggested for the UK.

Coal consumption in electricity generation is also relatively high in Denmark, but here too, the government has announced its intention to reduce its reliance on the fuel. In Denmark close to 30% of electricity was generated by renewables in 2013; the country indicated in 2011 its intention to phase out fossil fuel generation completely by 2050, with a 40% emissions reduction target for power generation set for 2020 (relative to 1990).[69]

In other countries such as France and Sweden, decarbonisation of the electricity supply system is already well advanced, with France being reliant on nuclear generation, and Sweden using large amounts of biomass within its generation mix.

In the case of heat - unlike electricity - there is no grid. In considering the marginal power source for heat within life cycle assessment, some authors have therefore argued that local conditions are of greater relevance and have used the average mix of fuels for a region within the country when considering the impacts of a specific plant.[70] Across Europe, natural gas is also widely used for heating as well as electricity; a German study in 2011 suggested gas dominated the national heat mix, with the remainder being oil (a more carbon intense fuel than natural gas).[71] Most decarbonisation trajectories are heavily focused on electricity generation with much less discussion on heat.

4.3.3 Approach used to Model Incineration

Our analysis is based on the performance of an electricity-only incinerator, with a gross generation efficiency of 26% (equivalent to a net generation efficiency of 23%). CEWEP data suggests this is on the high-side for existing facilities in operation within Europe, but it will perhaps be on the low side for facilities that are under construction / in planning. Performance variations resulting from the avoided generation of different fuels for electricity are also considered. To calculate the direct emissions to air from the facility, the composition data previously presented in Table 4 -20 is taken together with data on the carbon contents of waste materials.[72]

4.4 Mechanical Biological Treatment

A variety of MBT processes operate in Europe. Impacts depend on the treatment steps included within the process.

- The mechanical step involves the separation of recyclate from the rest of the residual stream, leading to the climate benefits previously described in Section 2.2. In many systems, some subsequent separation of different residual streams also occurs – either to separate out an organic fraction which then undergoes biological treatment, and/or to separate out different fuel streams.
- The biological treatment step can be either aerobic or anaerobic. In the case of the former, the objective may be to dry the material, increasing its calorific value prior to the material being used as Refuse Derived Fuel (RDF), or to stabilise it, such that the environmental impact in landfill (in terms of the amount of methane generated) is decreased. Energy generation is also the objective of the anaerobic-based systems.
- Depending on the nature of the previous steps, then, different fractions may result:
    - o One or more fuel streams may be produced and used in an incinerator, gasifier, or to displace coal in a cement kiln;

68 The Oxford Institute for Energy Studies (2014) The New German Energy Policy: What Role for Gas in a De-carbonization Policy?
69 See http://www.danishenergyassociation.com/Theme/Decarbonisation.aspx; OECD (2013) Renewable Energy: A Route to Decarbonisation in Peril?
70 Bernstad A, La Cour Jensen, Aspegren H (2011) Life Cycle Assessment of a Household Solid Waste Source Separation Programme: A Swedish Case Study
71 Buhle L, Stulpnagel R and Wachendort M (2011) Comparative life cycle assessment of the integrated generation of solid fuel and biogas from biomass (IFBB) and whole crop digestion (WCD) in Germany, Biomass and Bioenergy, Vol 35(1), pp363-373
72 The latter is taken from the waste model; see: Eunomia and CRI (2014) Development of a Modelling Tool on Waste Generation and Management: Appendix 6 Environmental Modelling

o There may be a compost-like product which is used for land remediation purposes;

o Stabilised material may be landfilled.

Factors influencing the performance of stabilisation-based systems will differ, to a certain extent, to those affecting the RDF-based systems. However, the performance of all such systems is determined in part by the efficiency and operation of the mechanical separation technology employed within the MBT, which not only influences the amount of recyclate and the quality of the recyclate stream produced, but also affects the quality - and therefore the economic value - of the product streams. The associated emissions impacts therefore flow from this.

Emissions occurring directly from the mechanical separation and biological treatment elements of the process are typically relatively insignificant. For the RDF-based systems, the major impacts occur from the use of the RDF as a fuel and the associated energy generation benefits. There are dual benefits from the recycling of materials: both in terms of emissions savings from the actual recycling as well as emissions reductions occurring from the removal of the fossil-CO_2 containing plastics from the fuel stream. On the other hand, stabilisation processes offer an opportunity to reduce landfill emissions. Depending, therefore, on the nature of the process being operated, different elements from the previous discussion on the factors affecting landfill / incineration / recycling processes will be relevant for MBT processes.

The following types of MBT facility were included in Eunomia's European Waste model, reflecting the most commonly used approaches:

- The stabilisation of the degradable fraction to reduce impacts from landfilling;
- Biodrying to produce a fuel subsequently used in an incinerator; and
- Processes that use AD to treat the biodegradable element of residual waste.

Impacts of these types of MBT systems are considered in the next section.

4.5 Summary of Residual Waste Treatment Impacts

Taking into account the above discussions on assumptions, residual waste treatment impacts are summarised in Table 4 -25. Data is presented for one tonne of residual waste, modelled as per Section 4.1. Impacts are presented for landfill and incineration. The impacts for gasification facilities are likely to be similar to that of the incineration facilities for the reasons discussed in Section 4.3.1.2, and so these have not been separately presented here.

The table here also includes – for comparison purposes - indicative impacts for the MBT systems which were not included within the analysis provided in the main report. The data presented in the table indicates that these may be expected to slightly perform better than the best performing landfill systems, although performance is not as good as the situation where an incinerator generates electricity where the avoided source of energy is electricity generated using coal. The performance of these systems is, then also well within the spectrum of performance for the other residual waste treatment systems.

The data in the above table considers only incinerators generating electricity. As has been indicated in the prior discussion, in some northern European countries (including Denmark, Germany, the Netherlands and Sweden), incineration facilities generate heat as well as, in some cases, electricity. Here the overall energy generation efficiency of such facilities is considerably in excess of the performance of the facility which has been used to model the impacts in Table 4 -25.

The effect of these increased efficiencies on the climate change impact is still, however, highly dependent upon the source of energy that is assumed to be displaced. This is more difficult to determine for heat, as was discussed in Section 4.3.2. In many cases, the displaced heating fuel is likely to be gas; in this case, the performance of the facility is unlikely to be better than the incinerator generating electricity where the avoided fuel is coal (although

Table 4-25: Summary of Residual Waste Treatment Impacts

		Climate change impacts, kg CO2 equivalent per tonne of residual waste	
		Excluding biogenic CO2 emissions	Including biogenic CO2 emissions
Landfill	Gas Capture 20%	506	1,000
	Gas Capture 50%	202	721
	Gas Capture 70%	-6	535
Incineration (generating only electricity)	Avoided electricity source - coal	-296	252
	Avoided electricity source - gas	52	528
	Avoided electricity source - wind	288	821
MBT	Stabilisation	-25	294
	Biodrying	-24	490
	AD-based	-30	315

such plant would do better than the electricity-only facility where gas use was avoided). For a more significant climate change benefit than that seen in Table 4 -25, an incinerator generating heat would need to be offsetting the use of coal or oil as a heating fuel, a situation that is becoming relatively less common in Europe. By contrast, in many cases in Sweden, the use of waste as a heating fuel would be displacing the use of biomass. The climate change impact in this case would depend on the sustainability of the biomass fuel the use of which is being avoided by the incinerator, but if the fuel is from sustainably managed feedstocks, impacts could be similar to those seen for the electricity-only facility where wind was the avoided fuel source.

Section 4.2.2 discussed the GWP of methane. As this increases, so the impact associated with landfill rises. Where the results are considered with a 20 years GWP, the impact of landfill would be considerably greater than that shown here. However, the impact of this rise on the MBT technologies – such as the stabilisation technologies which seek to mitigate the worst impacts of emission from landfill - would be much less significant.

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